School of Engineering

Pontifical Catholic University of Chile

Co-controls Benefits Analysis for Chile

Preliminary estimation of the potential co-control benefits for Chile

COP-5 Progress Report

Luis A. Cifuentes
Enzo Sauma
Hector Jorquera
Felipe Soto

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Corresponding Author: Luis A. Cifuentes, lac@ing.puc.cl

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1. Introduction

There is no doubt than human activity is responsible of the increasing atmospheric concentrations of greenhouse gases (GHG), including carbon dioxide (CO2), methane (CH4) and nitrous oxide (N2O). The main activities responsible are fossil fuel combustion, which has grown at a rate unprecedented in human history, and changes in land use and agricultural practices. In the absence of control of emissions of these gases, their atmospheric concentrations throughout the next century will rise to levels that may induce changes in the climate of the earth. The Intergovernmental Panel on Climate Change (IPCC) estimates that human-induced climate change will increase surface temperatures by about 2°C by the year 2100 (Houghton et al. 1996), although many uncertainties exist about this estimate.

The climate change protocol signed in the third conference of parties in Kyoto in December 1997 set goals for emissions reduction for countries included in Annex I, that includes only developed countries. Non Annex I countries, mainly developing countries, do not need to abide by any emission reductions. The protocol set up an emissions trading framework that would allow countries (mainly Annex I) to invest in GHG reduction projects in other countries, and share part of the emissions credits. The implementation of such schemes, like "Joint Implementation" and "Clean Development Mechanisms" were widely discussed at the last conference of parties held in Buenos Aires, in November 1998.

In order to stabilize the global emissions of GHG, it will be necessary for all countries, including developing ones, to make reductions in their emissions. However, developing countries shall make the most progress in reducing the growth of their greenhouse gas emissions by implementing measures that are consistent with their development objectives and that provide near term economic and environmental benefits.

Within the existing framework, it is not clear for a developing country if it is beneficial to enter voluntarily in an emission reduction scheme. Our previous results for an analysis for Chile (Montero, Cifuentes, and Soto 1999) shows non-conclusive results, with the economic convenience depending heavily in the initial emissions baseline assigned to the country.

While many developing countries have conducted extensive analysis of possible greenhouse gas measures, little attention has been given to full characterization of the more immediate environmental and health benefits that would result from these measures.

Understanding these benefits has been a critical gap in past efforts to help developing countries estimate the cost-effectiveness of greenhouse gas mitigation. Improving a country's understanding of the scope and potential magnitude of these direct public health benefits can help develop better policy recommendations considering the full impact of adopting alternative climate change mitigation policies.

Figure 1-1 Short and long term social benefits derived from control of GHG emissions

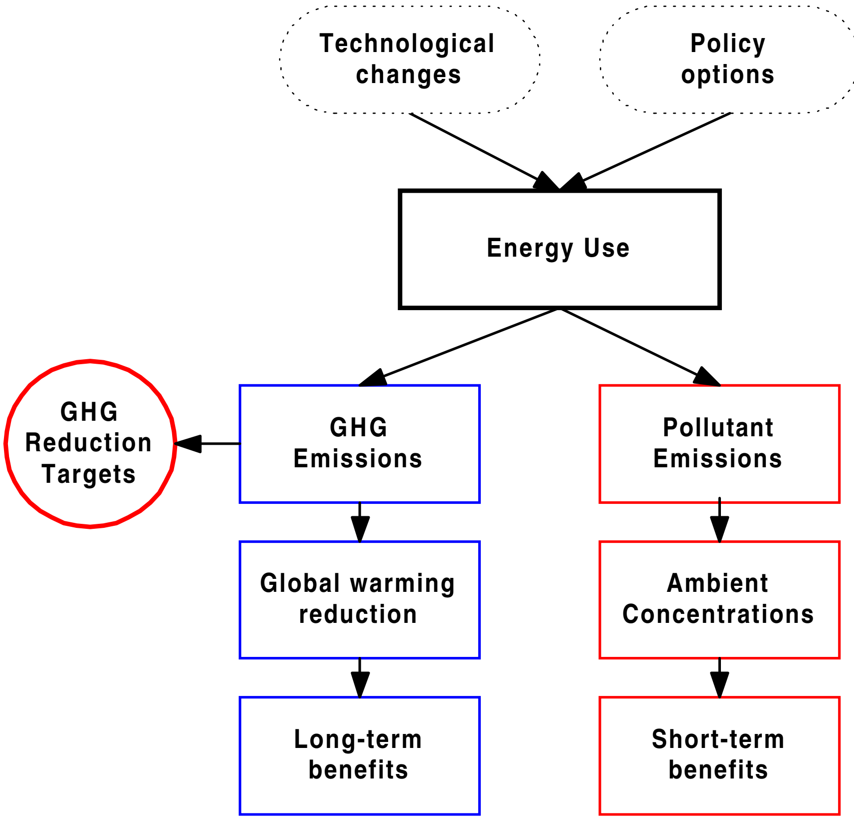


Figure 1-1 shows a schematic vie of the potential social benefits resulting from measures aimed at reducing GHG emissions. Technological and policy options aim at reducing energy use to achieve the target in GHG emissions reductions. Global warming reductions lead to long-term benefits, such as reduced extreme weather events, sea level rise, and communicable diseases spread, among others. However, these benefits are uncertain (at the very least to the extent that global warming itself is uncertain) and, from the point of view of a single country, global warming reduction is a public good. Therefore, it is not strange than developing countries are more preoccupied with more pressing, local, immediate environmental and human health needs, such as control of air and water pollution, and control of infectious diseases, than with global, long-term problems, such as global warming.

However, the right side of the scheme shown in Figure 1-1 shows a path to more certain health benefits from GHG reductions. The same combustion processes that lead to emissions of GHG also produce local and regional pollutants, like particulate matter (PM), sulfur dioxide (SO2) , and nitrogen oxides (NOx), which have potential effects on human health. Thus, a reduction in GHG emissions may produce concomitant reductions in these pollutants, and may lead to reduced short-term damages from air pollution, such as health effects, vegetation effects, materials damages, and visibility impairments. Of all these effects, health damages are probably the most important ones. If properly considered, these short-term health benefits may allow for implementation of policy measures that would otherwise have not been taken. If these health benefits are significant, they may even allow for "no regrets" GHG abating measures, in which taking immediate action to reduce GHG will be justified, even if later it is discovered that global warming is not as big a threat as considered today.

In this report, we estimate the potential co-control benefits for Chile. The analysis is preliminary. It is based on an emissions scenario developed previously in a previous study, and draws on several pieces we have been working independently in the past. Although the analysis is subject to several limitations, we believe its results show the potential for co-control benefits of GHG mitigation in Chile, and can be useful to Chilean policy makers. This is the first step in a year-long study aimed at identifying the full potential of these secondary GHG benefits.

1.1 Organization of this report

The report is organized as follows. Chapter 2 presents the scenarios used to estimate the potential co-control benefits, taken from a previous study, and presents an estimation of the human exposure of the Chilean urban population to fine particulate matter. Chapter 3 presents the methodology used to derive the co-control benefits. Chapter 4 presents the main results, and finally, we present our conclusions on Chapter 5.

2. Scenario Development

We have considered two emissions scenario: the Business-as-usual (BAU) scenario, in which no GHG mitigation measures are taken, and a Climate Policy (CP) scenario, in which some measures are taken to reduce emissions.

For this preliminary assessment, we have relied on the results obtained in a recent study commissioned by the Chilean Environmental Commission to the Research Program on Energy (Programa de Investigaciones en Energía) of the University of Chile (PRIEN 1999).

The study projected the emissions for several greenhouse gases, including carbon dioxide (CO2), nitrous oxide (N2O) and methane (CH4) and several primary pollutants, including sulfur dioxide (SO2), nitrogen oxides (NOx) and non-methane hydrocarbons (NMHCs). The study is based on an engineering, bottom-up approach, considering technological measures like efficiency improvements, and fuel switching to obtain emissions reductions.

For the base case, policies that are currently in place and those which are scheduled to be apllied were considered. In particular, all the measures of the Decontamination Plan for the Metropolitan Region that are scheduled to be implemented were considered (Comisión Nacional del Medio Ambiente 1997), as well as the future investments in infrastructure contained in the national strategic plan (MOP 1997).

The most relevant assumptions considered for the projection of the base scenario from 2000 up to 2020 are:

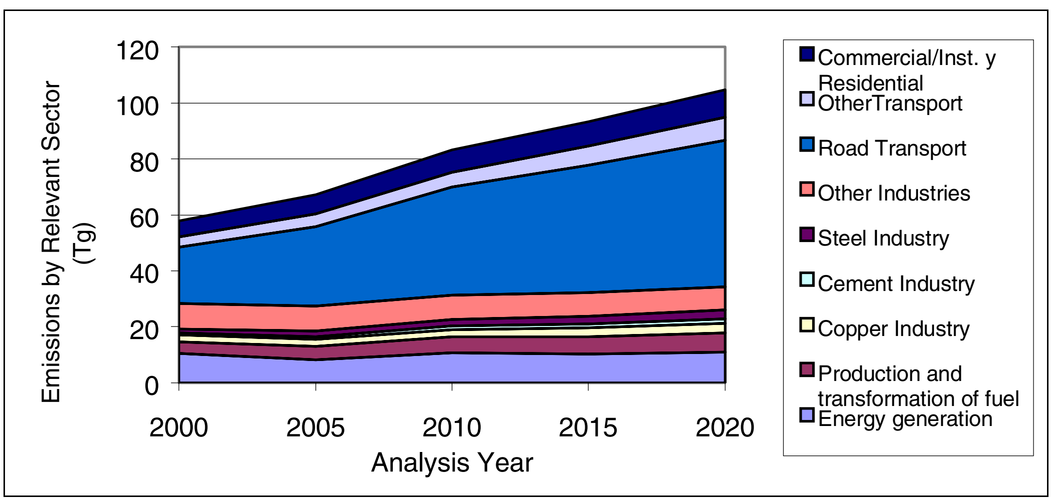
- an average annual GDP growth of 5% for the whole period of analysis (2000 - 2020).
- An urban population increase of the 1.4 % annually during the period.
- An constant rural population of around 2.5 million people.
- No substantial variations in the prices of energy.

2.1 GHG emissions

The Business-as-usual scenario projected the emissions of CO2, CH4 and NO2 for the years 2000 to 2020, with 5-year intervals. Based on the Global Warming Potentials (GWP) factors recommended by IPCC (IPCC 1996) we computed the CO2 equivalent emissions for each year.

The emissions were projected for the different sub-sectors of the economy. Appendix I presents the emissions of each pollutant in detail for each sub-sector. We aggregated the data into the most relevant sub-sectors, as shown in the next figure.

Figure 2-1 CO2 equivalent emissions by sector, for the BAU scenario



Source: (PRIEN 1999).

It can be observed in the figure that the CO_2-equivalent emissions grow 81,3% during the period 2000-2020 for the BAU scenario. This high growth is explained mainly by the explosive growth of the emissions in the road transport sector, as is clearly seen in following table.

Figure 2-2 CO_2-equivalent emissions relative to emissions in the year 2000, for the most relevant subsectors of the economy, for the BAU scenario

Relevant Sector	2000	2005	2010	2015	2020
Electricity generation	1.00	0.79	1.04	0.99	1.06
Production and transformation of fuel	1.00	1.17	1.37	1.48	1.62
Copper Industry	1.00	0.95	0.97	1.18	1.31
Cement Industry	1.00	1.40	1.77	2.08	2.33
Steel Industry	1.00	1.31	1.63	1.94	2.25
Other Industries	1.00	0.98	0.96	0.94	0.92
Road Transport	1.00	1.42	1.93	2.27	2.61
OtherTransport	1.00	1.25	1.50	1.88	2.28
Commercial/Inst. y Residential	1.00	1.20	1.39	1.53	1.72
total	1.00	1.16	1.44	1.62	1.81

Source: aggregation of data from (PRIEN 1999).

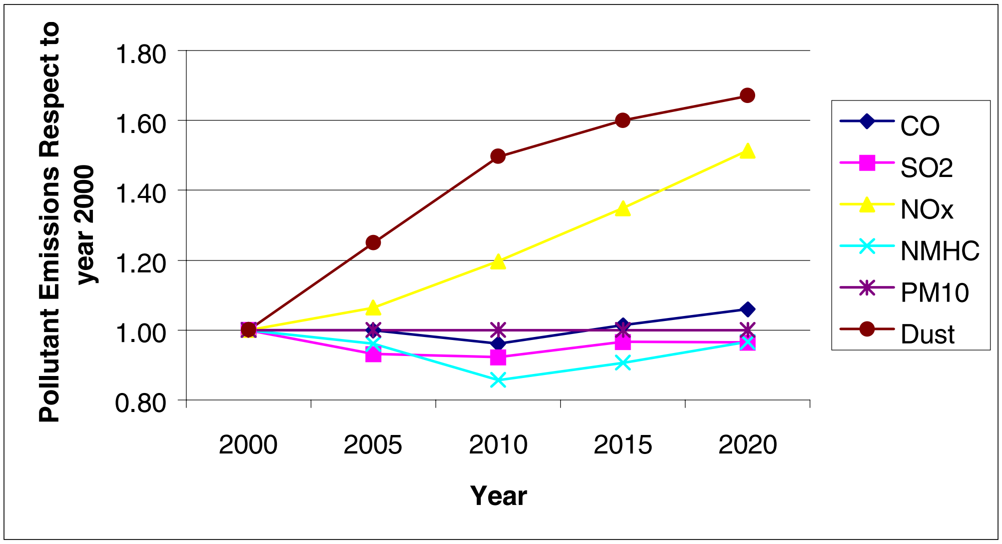
2.2 Emissions of primary pollutants

As mentioned above, the emissions inventory (EI) considers the emissions of CO, SO2, NOx and NMHC. Besides these pollutants, two other pollutants are important to estimate the change in particulate matter concentrations: direct emissions of PM_{10} from combustion sources, and emissions of resuspended dust from the operation of motor vehicles. Since there was no reliable estimation for those emissions, we had to project their growth based on indirect, available data.

In this way, we assumed that resuspended dust emissions would increase in proportion to the use of energy in the transportation sector. However, to consider the impact of increased paved surfaces in urban roads, we decreased the emissions by 2% annually. This also represents the implementation of street cleaning programs in major cities, especially in the Metropolitan Region.

The estimation of the growth of direct PM_{10} emissions from combustion sources is more complicated. As a first approximation, it depends on the amount of each fuel used, and the emissions factors associated to each fuel in each sub-sector. At this moment, we do not have the detailed energy consumption by fuel for each sub-sector. Due to this limitation, we have assumed that PM_{10} emissions will not vary between scenarios. This is a conservative assumption, since clearly PM_{10} emissions should be smaller for the CP scenario. However, owing to our approaches to estimate the changes in $PM_{2.5}$ concentrations (basically based on all pollutants – not only on PM_{10}) , this assumption is less critical than it may appear at first sight. If and when we obtain detailed data on past and projected fuel consumption figures by economic sector, we will be able to refine this estimate.

Figure 2-3 Relative emissions of primary pollutants with respect to year 2000, for the BAU scenario



The figure emphasizes the relatively high growth of the resuspended dust emissions, and of NO_x, both coming from the explosive growth of the transportation sector, as was explained before. On the other hand, the emission levels of the others primary pollutants are approximately constant during the period 2000-2020, due to increases in energy efficiency and also due to the steady introduction of natural gas.

2.3 Potential of emission reductions in the Control Policy Scenario

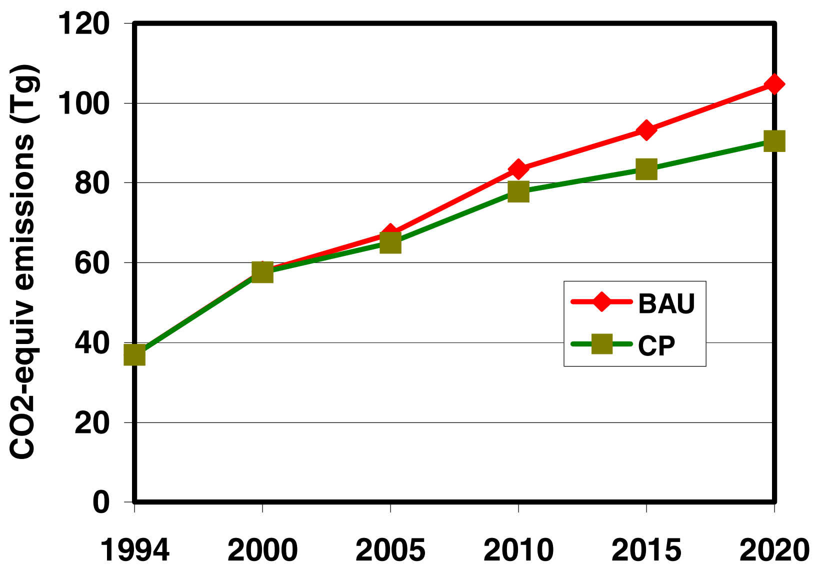
As mentioned before, we have considered as climate policy scenario, the mitigation scenario developed by the Research Program in Energy of the University of Chile (PRIEN 1999). This mitigation scenario was developed following the bottom-up (or engineering) approach, considering the introduction of newer, more efficient technologies and computing the incremental cost and emissions reductions. Since the objective of the study was to estimate emissions reductions that could be achieved through "no-regrets" implementation of technologies, the adoption and rate of penetration of the technologies was determined such that they would represent a net cost saving to the user. New technologies were considered for all sectors: residential, commercial, industrial and transport. Technologies considered in the residential/commercial sector included for example improved appliances, compact-fluorescent lamps. In the industrial sector, the main technologies were newer, more efficient, electric motors, and the increased use of co-generation. In the transport sector the main mitigation measures are mode switching to cleaner means of transportation, and improvements in the fuel efficiency of the existing means of transport.

Due to the way the mitigation scenario was constructed, we can assume that the mitigation costs would be negative or close to zero, but the associated reductions in greenhouse gas emissions are relatively small.

Since we are interested in a preliminary estimation of the co-control benefits, this may not be a serious limitation. We are just computing the co-control benefits for the first mitigating measures. We are not going up the marginal cost of mitigation curve.

Appendix I presents the detailed emissions of each pollutant, in the climate policy scenario, for the most relevant sub-sectors of the economy.

Figure 2-4 CO2 equivalent emissions for the BAU and CP scenario



In the previous Figure, note that in the CP scenario, the CO_2 equivalent emissions will grow only 57% during 2000-2020, that is, a decrease of the 13,5 % from the business-as-usual scenario. The main causes for this reduction are lower emissions in the Road Transport and Commercial/Institutional and Residential sub-sectors, as is clearly shown in following table.

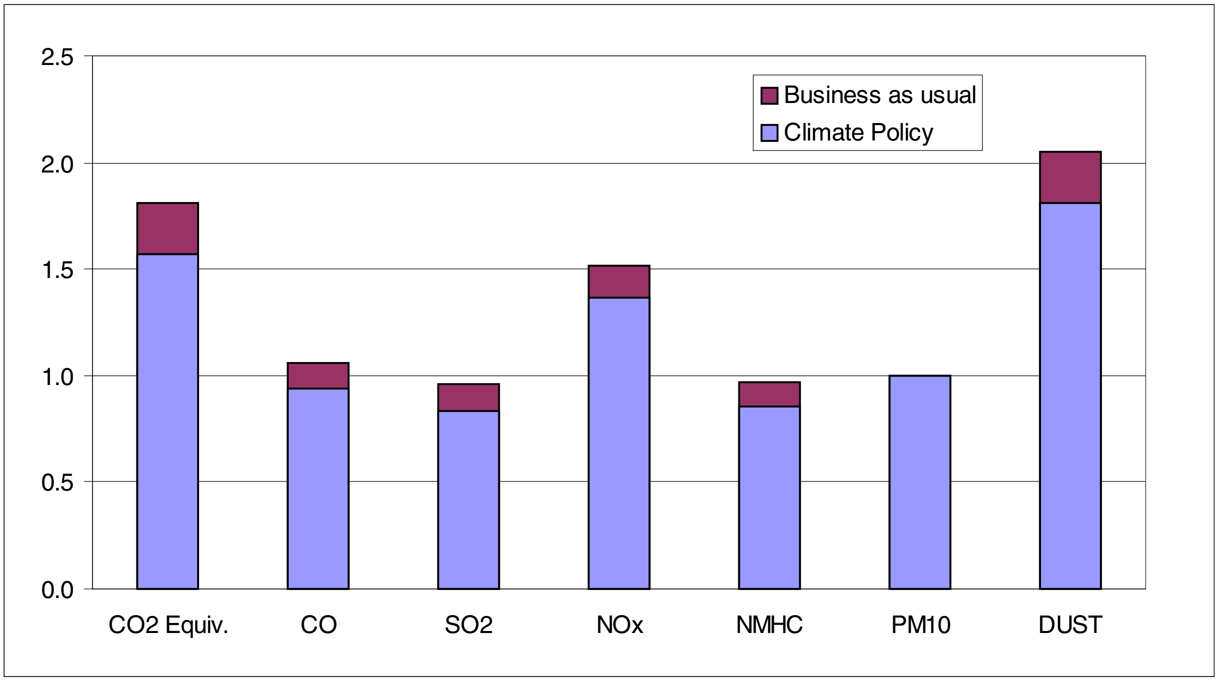
Figure 2-5 CO_2-equivalent emissions in the CP scenario as fraction of BAU emissions, for the most relevant subsectors of the economy.

Relevant Sector	2000	2005	2010	2015	2020
Electricity generation	1.00	0.99	0.94	0.86	0.85
Production and transformation of fuel	1.00	0.99	0.97	0.94	0.92
Copper Industry	1.00	0.98	1.04	0.93	0.90
Cement Industry	1.00	1.00	1.00	1.00	1.00
Steel Industry	1.00	0.90	0.84	0.80	0.77
Other Industries	1.00	0.95	0.90	0.85	0.80
Road Transport	1.00	0.98	0.94	0.89	0.85
OtherTransport	1.00	1.00	1.01	1.01	1.01
Commercial/Inst. y Residential	1.00	0.88	0.87	0.85	0.83
Total	1.00	0.97	0.93	0.89	0.86

The Steel Industry and the Other Industries sub-sectors show the biggest reductions from the BAU scenario, of 23% and 20% respectively, for the year 2020. However, the absolute emission levels of these sub-sectors do not have a big influence on the total emissions of the country.

On the other hand, it is interesting to analyze the ratio between emissions under the climate policy scenario (CP) and under the Business-as-usual scenario (BAU), for the primary pollutants.

Figure 2-6 BAU and CP emissions for year 2020 relative to year 2000 , for all pollutants



The figure does not include a reduction in the PM10 because, as we mentioned above, we are still working to identify a better emissions estimation method for this pollutant.

2.4 Human exposure to air pollution in Chile

Fine particulate matter ($PM_{2.5}$) was used as a sentinel pollutant to estimate the extent of human exposure to air pollution in Chile. We choose to concentrate on $PM_{2.5}$ because recent studies conducted in the US (Schwartz, Dockery, and Neas 1996) as well as our own results (Cifuentes, Vega, and Lave 1999) show that the fine fraction of particulate matter is responsible for mortality effects.

Chile has a relatively widespread ambient particulate matter pollution problem. Santiago, the capital of Chile, is one of the world's most polluted cities by particulate matter. Regular daily measurements of $PM_{2.5}$ and PM_{10} using dichotomous samplers began in 1988 in five stations across the city, and has continued since then. The original network was expanded in 1997 with a new network of eight monitoring stations. The rest of the country is not been so well documented. Regular monitoring is not currently conducted in any other city, except a few localities close to copper smelters, which have been declared saturated zones, where the law mandates regular monitoring to ensure that ambient air quality levels are improving (or at least not worsening).

The next table shows the main cities in Chile, with its population and its estimated PM concentrations. The cities which have particulate matter measures, as $PM_{2.5}$ or PM_{10} comprise a total of 7.8 million people, or about 63% of the projected urban population of Chile in the year 2000. The Metropolitan Region surrounding Santiago represents 72% of this population, and 45% of the total urban population of Chile. For cities that do not have measurements of $PM_{2.5}$, we have estimated them based on the average ratio of $PM_{2.5}$ to PM_{10}. For those with no measurements at all, we have assumed a level equal to the cleanest city measured, equal to 13.5 ($\mu g/m^3$). This assumption can be conservative or not. Some industrial cities like Talcahuano probably have a higher concentration, and smaller cities may have lower ones.

Figure 2-7 Main cities in Chile with measurements of particulate matter concentrations in 1998

City	Urban Population persons	Particulate Matter Concentration PM2.5 ug/m3	PM10 ug/m3	PM2.5 equiv ug/m3	Source
Rancagua	200,771	42.7	73.2	42.7	(2)
Santiago MR	5,671,689	38.8	82.4	38.8	(1)
Huasco	6,786		76.0	36.7	(3)
Iquique	168,383	36.0	63.4	36.0	(2)
Temuco	235,361	35.3	66.9	35.3	(3)
Copiapo	109,739		71.5	34.5	(2)
Valparaiso	314,848	26.7	77.6	26.7	(4)
Concepcion	365,228		54.1	26.1	(2)
Chañaral	14,743		52.0	26.0	(3)
Arica	180,313		48.6	23.5	(4)
Viña del Mar	339,305	22.4	56.9	22.4	(2)
Antofagasta	253,538		42.8	20.7	(4)
Caldera	13,122		28.0	13.5	(3)
Talcahuano	274,877			*13.5*	
Talca	179,791			*13.5*	
Chillan	162,907			*13.5*	
Other	3,959,610			*13.5*	
Total	**12,250,240**	31.8	60.0	24.6	

Sources:

(1) (SESMA 1999)

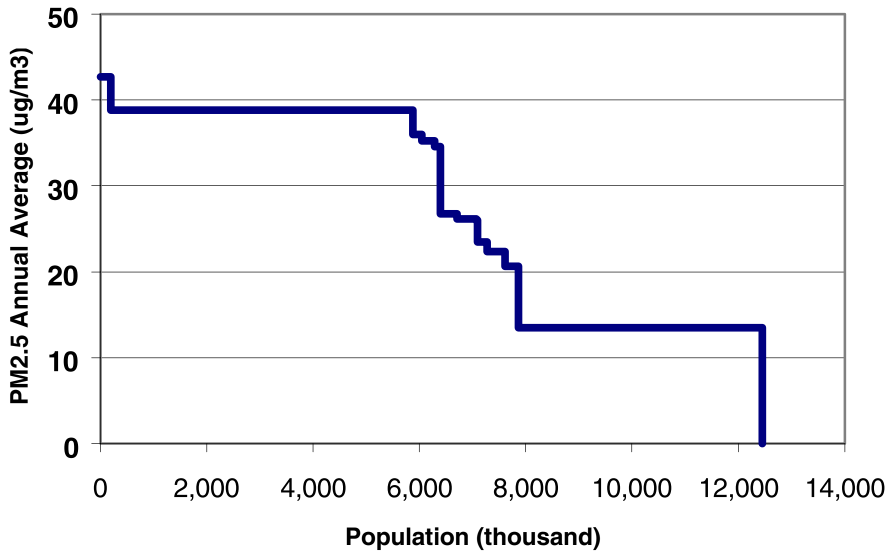
(2) (Cosude 1999)

(3) (CIMM 1998)

(4) (Gredis 1999)

The next figure shows an estimation of the exposure level for all the urban population in Chile. The figure underlines the relative importance of the Metropolitan Region of Santiago in the total exposure of Chilean population. The total population exposure in the year 2000 will be 344.5 Million person*(µg/m^3) of $PM_{2.5}$, of which 66% correspond to the Metropolitan Region.

Figure 2-8 Projected exposure of the Chilean urban population to PM2.5 in the year 2000



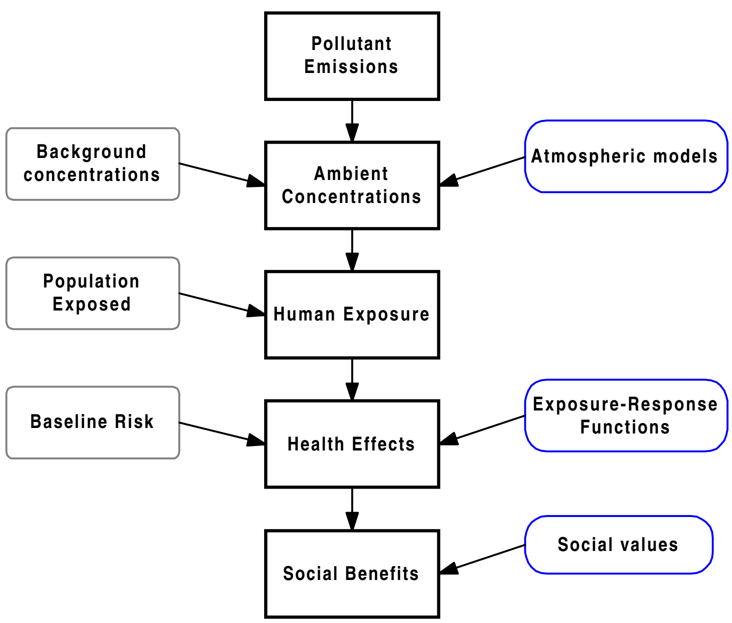
3. Methodology

There are several levels at which the co-benefits of GHG analysis can be conducted. The most detailed would be an individual mitigation measure analysis, in which the changes in pollutant emissions associated to each policy or technological measure are estimated, and linked to a change in health effects of the population. This requires a great deal of data. The impacts of different mitigating measures are likely to vary according to the location and duration of the reduced emissions, the population density close to the sources, the prevailing metrological conditions, etc. For this preliminary analysis we took a more global approach, conducting the analysis at an aggregate level for the whole country.

The first step to estimate the short-term health benefits it to link each policy or technological measure to the reduction in emissions pollutants. As explained in the previous section, the changes in emissions were taken from a previous study.

Once the changes in pollutant emissions have been assessed, it is necessary to link them to changes in ambient concentrations, population exposure, health effects and social benefits, using the Damage Function Method, showed schematically in the next Figure.

Figure 3-1 Framework to estimate the social benefits of a reduction in emissions of primary air pollutants



To estimate the potential health benefits, we assembled a database of the current exposure of the Chilean urban population to particulate matter. Several studies conducted or commissioned by the National

Commission of the Environment (CONAMA) have measured particulate air pollution in cities that comprise almost half of the Chilean population living in urban areas.

The changes in ambient concentrations of particulate matter were estimated using two methods, based on source apportionment, and in statistical associations between atmospheric pollutants. Although both methods were developed using data specific for Santiago, we applied their results to the whole country. This assumes that the atmospheric processes for the rest of the country are similar to Santiago's, which is a crude assumption, that may be justified for the northern half of the country (similar predominance of anticyclonic conditions and little if any rainfall), but clearly does not apply well to the southern half of the country, where more precipitation and passage of cold, low pressure fronts tend to dilute pollutants better than at Santiago. Unfortunately, due to limited data, this is the only option available to us at this time.

With the projected ambient concentrations for each policy scenario, we computed the population exposure in each year. Drawing on data of a previous study in which we estimated the social losses due to particulate air pollution in Santiago, we estimated the health damages for the CP and the BAU scenario, obtaining the social benefits as the difference of the two. In the next sections we describe in detail the methods used in each step.

3.1 Changes in air pollutant concentrations due to changes in emissions of primary pollutants

This step is a crucial part of the method linking emissions of primary pollutant to social losses. For a detailed analysis, it should rely on atmospheric dispersion models, specifically in models that incorporate the complex set of chemical reactions occurring in the atmosphere. None of those models are available for Chile at this time. Some models are in the development process for Santiago, but are non-operative yet. For instance, CONAMA is working in close cooperation with the Swedish Meteorological Institute to establish a regional air pollution dispersion model for the V, VI and Metropolitan Regions (http://tralka.dcc.uchile.cl/match/). This model will provide a more accurate account of transport and fate of airborne pollutants. The preliminary results are scheduled to be released late this year or early in 2000. In principle, the outcome of the model could be used to develop a Source-Receptor matrix (SRM) for Central Chile, although we are not sure if budget and human resources constraints will allow CONAMA to do so.

For this preliminary analysis, we estimated the impacts of emissions changes on PM concentrations based on two aggregate methods.

3.1.1 Method 1: Use of a box model to develop emission concentration relationships.

A highly simplified methodology was used to estimate the future impacts of PM_{10}, $PM_{2.5}$ and coarse fractions. The starting point is the Eulerian Box model approach that reads

$$\frac{dC_i}{dt} = \frac{q_i}{H(t)} + R_i - \frac{V_{d,i}}{H(t)} C_i + \frac{u(t)}{\Delta x}\left(C_i^U - C_i\right) + \frac{(C_i^A - C_i)}{H(t)} \frac{dH}{dt} \qquad (1)$$

where C_i is the pollutant concentration ($i = PM_{10}, PM_{2.5}$, etc.), $H(t)$ is the mixing height, q_i the surface emission in the box, R_i the net production rate by chemical mechanisms, $V_{d,i}$ the deposition flux (dry and wet) at the ground surface, $u(t)$ the average wind speed in the box and the superscripts U and A stand for upwind and aloft advected concentrations, respectively. The rightmost term on the right hand side of (1) is only applicable whenever the mixing height is rising, that is, from sunrise until early afternoon (Seinfeld and Pandis 1998).

The above equation describes mathematically the concentration of species above a given area, accounting for emissions, chemical reactions, removal, advection of material in and out of the airshed and entrainment of material during growth of the mixed layer. The strongest assumption is that the corresponding airshed is well mixed.

Since only 24 h averages are being routinely measured for PM_{10}, $PM_{2.5}$ and coarse particles in Chile, reordering and integration of equation (1) produces the following equation for the 24 h concentration - emission relationship

$$< C_i > = < C_i(0) > + \tau\left(< R_i > + < B_i >\right) + \tau < \frac{1}{H} > \left(\alpha q_{CO} + \beta q_{SO2} + \gamma - \left[C_i\right] < p >\right) \qquad (2)$$

where the angular brackets mean 24 h averages and τ is a time scale of combined advection transport and deposition removal given by

$$\tau^{-1} = \frac{< u >}{\Delta x} + < V_{d,i} > < \frac{1}{H} > \qquad (3)$$

this quantity is only dependent upon meteorology and surface conditions. This time scale surely has a seasonal variation, for during fall and winter, the poorer ventilation conditions decrease both $<u>$ and $<1/H>$ in the above expression.

The terms $<R_i>$ and $<B_i>$ represent the contributions to the daily average concentration coming from secondary aerosol production by chemical mechanisms and by advected particles from upwind sources and from aloft, respectively.

The rightmost parenthesis in equation (2) is the decomposition of the net surface emission of particulate matter into the following fractions:

a) Primary emissions from vehicle exhaust, which must be proportional to the concomitant CO emission. This term stands for directly emitted PM from mobile sources (the dominant source of CO).

b) An emission coming from stationary sources that ought to be proportional to SO_2 emissions, which in turn are largely dominated by stationary sources (industrial and domestic).

c) An emission that cannot be ascribed to any of the above combustion processes; this emission corresponds to mechanisms such as resuspended street and surface dust, tire and brake wear, etc. We will denote this contribution to ambient concentrations by the symbol $<C_{RSP}>$, because this contribution is dominated by PM resuspended by transportation.

d) A negative contribution coming from wet deposition; the square brackets mean average concentration of the pollutant in the aqueous phase. The $<p>$ symbol represents average precipitation intensity (mm/h) recorded the same day. This contribution to ambient concentrations will be denoted by the symbol $<C_{WD}>$.

In order to validate the above model, data gathered at Santiago for the fall and winter seasons from 1990 to 1994 were used to fit the model (in some cases, data from 1995 and 1996 were used to increase the database, as was the case in Station C). The air quality data came from the MACAM monitoring network, and included hourly measurements of CO, SO_2, and surface wind speed u plus daily measurements of PM_{10}, $PM_{2.5}$ and coarse fractions. A substantial amount of time was devoted to extracting 24 h averages of the different terms appearing in equation 2, considering missing values, analyzing partial scatter plots to detect outliers, and so on. For instance, the 24 h measurements for the high volume, dichotomous sampling started every day at 10 am (LST) and ended the next morning, after 24 h of consecutive sampling. An additional record in the database indicated the effective number of hours that actually happened at a given day; hence only samplings that lasted more that 18 h were considered representative for the analysis.

In practice, we cannot resolve separately the reaction, advection and constant terms occurring in equation (2), so all of them are collapsed in a constant, background term. That is, we only produce an estimate of the seasonal contribution of these three terms to the ambient air quality level. The working equation that reflects the emission concentration relationship is

$$< C_i >=< C_B > + \tau < \frac{1}{H} > \left(\alpha q_{CO} + \beta q_{SO2} + \gamma - [C_i] < p > \right) \qquad (4)$$

where $<C_B>$ is the aforementioned seasonal background term for fine, coarse and PM_{10} particulate matter.

3.1.1.1 Parameter Estimation and Model Validation.

Model parameters were obtained by using classical, linear regression analysis of equation (4) using the measured, daily concentrations of CO and SO_2 in place of the unobserved daily emissions; in this case, the meteorological factor $\tau<1/H>$ cancels out in the above equation and we simply regress $<C_i>$ using as predictands $<C_{CO}>$, $<C_{SO2}>$, $<1/u>$ as surrogate for $<1/H>$ and $<p><1/u>$ as surrogate of $<p><1/H>$. In this fashion, we could estimate the model parameters for stations A, B, C and D of the MACAM network, and for the three fractions: PM_{10}, $PM_{2.5}$ and coarse particles. In particular, we can estimate the different contributions to the total, ambient particle concentrations coming from

a) Advected, secondary and background levels

b) Directly emitted by combustion in mobile sources

c) Directly emitted by combustion in stationary sources

d) Deposited onto the ground

Table II-1 (Appendix II) gives the 1997 Emission Inventory for Santiago (EIS), as developed by CENMA (1997). From this table, the estimate for α for the PM_{10} model would be the ratio of PM_{10} emissions from mobile sources to total CO emissions, that is

$$\alpha = \frac{2730(ton/yr)}{244921(ton/yr)} = 0.011(g/g)$$

The coefficients for the CO concentration in the PM_{10} model have the values 10.25, 9.97, 19.57 and 7.78 at stations A, B, C and D, respectively, when CO is measured in ppm and PM_{10} in ($\mu g/m^3$) (see Table 2). In units of (g/g), the coefficients take the values 0.009, 0.0087, 0.017 and 0.0068 for stations A, B, C and D, respectively. All coefficients are significant ($p<0.05$). - The similar results among monitoring sites and their reasonable agreement with the value estimated above from the annual emission inventory for Santiago show that the box model is capable of reflecting these relationships among primary emissions. In addition, the coefficient for SO_2 was higher than the value estimated from the 1997 EIS. From Table 1 we would estimate a value of β given by

$$\beta = \frac{3175/ton/yr)}{21169(ton/yr)} = 0.15(g/g)$$

In this same units, the fitted values for β are 0.24, 0.21, 0.50 and 0.58, with all of them being significant ($p<0.05$) for stations A, B, C and D, respectively. All these fitted values seem to be rather high, according to the 1997 EIS. We currently ascribe this discrepancy to a strong linear correlation between SO_2 concentrations and secondary sulfate formation by chemical reactions. That is, the β coefficient picks up part of the reaction term that is controlled by SO_2 concentrations, and this is a statistical artifact caused by the high correlation between $<C_{SO2}>$ and $<R_i>$ that cannot be circumvented within our modeling framework. In other words, the EIS data gives the ratio: (direct PM emissions / SO2 emissions), whereas the model fit gives ((direct PM + secondary sulfate) / SO2 emissions) .

There is an additional confounding factor because there is evidence of potential calibration problems at some of the stations during 1989-1994. More recent data (for 1997-1998) shows evidence of improved calibration. Calibration issues will be followed up in later stages of this project.

Nevertheless, secondary nitrate formation is currently neglected by the method, and is included within the background, constant term. The high correlation between CO and NOx concentrations led to linear models where NOx was not a significant predictand, so we had to discard NOx from the group of input variables. This issue ought to be revisited later in this work to come up with refined estimates.

In addition, the intercepts on the lineal regression equation produce estimates of the background levels of PM_{10}, $PM_{2.5}$ and coarse fractions. This is relevant information to be used in the estimation of future concentration impacts. From Table II-2 (Annex II) it can be seen that background levels of PM_{10}, $PM_{2.5}$ and coarse particles are around 45, 27 and 18 ($\mu g/m^3$), respectively. These three values compare very well with the <u>total</u> measurements made by (Artaxo 1998) at Buin, a rural site 35 km south of Santiago considered representative of upwind, background values for the greater Santiago area. The values reported by Artaxo *et al* in the winter 1996 campaign were 52, 29 and 23 ($\mu g/m^3$), for PM_{10}, fine and coarse particles respectively. The major difference lies in the coarse fraction, but (Artaxo 1998) measured $PM_{2.0}$ as fine fraction, thus explaining their larger estimates of the coarse particle background.

3.1.1.2 Projection of Future Impacts.

From the previous results, the working equation to estimate future concentrations under new emission scenarios is obtained from (4) in the following way

$$\frac{(<C_i> - <C_B> - <C_{RSP}> + <C_{WD}>)_1}{(<C_i> - <C_B> - <C_{RSP}> + <C_{WD}>)_2} = \frac{(\alpha q_{CO} + \beta q_{SO2})_1}{(\alpha q_{CO} + \beta q_{SO2})_2} \qquad (5)$$

Where 1 stands for 1994 and 2 for any future scenario. In addition:

a) The α and β coefficients estimated from the data fitting are valid only for the 1994 scenario; to estimate them in future scenarios the EIS for 1997 and 2005 developed for CENMA will be used to obtain future estimates of these two parameters. For instance, lower bounds for these parameters are the limits when all cars possess catalytic converters, all buses use CNG as fuel and all trucks possess BACT emission levels; upper bounds are the values estimated for 1994 in the model parameter estimation process.

b) It will also be assumed that Santiago will follow the same emission trend as the whole country in the PRIEN annual emission forecasts.

c) The estimates of contributions of resuspended dust and wet scavenged particle concentrations will be assumed to stay in the same ratio for all scenarios, that is

$$\frac{\left(< C_i >\right)_1}{\left(< C_i >\right)_2} = \frac{\left(< C_{RSP} >\right)_1}{\left(< C_{RSP} >\right)_2} = \frac{\left(< C_{WD} >\right)_1}{\left(< C_{WD} >\right)_2} \qquad (6)$$

this means that we assume that the emission factor for resuspended particles will stay the same; in other words, the proportion of ambient concentration will remain the same fixed percentage of the total concentration. Given the uncertainties in estimating this type of emission factor, we consider the above approximation reasonable; for instance, (Venkatram 1999) have reported estimates for this emission factor between 0.1 and 10 g/VKT, (VKT are the total kilometers travelled by all vehicles in a given period) for a metropolitan area.

d) The proportion of particles that are deposited by wet mechanisms is assumed to be the same as the values computed from the regression analyses: about 1 to 2% for most of the fractions. This means that, at least for Santiago, these quantities can also be incorporated in equation (5) as fixed proportions of the total, average concentration $<C_i>$ therein.

3.1.1.3 Results of the Simulated Scenarios.

In order to simulate impacts for the BAU and CP scenarios, the following specific assumptions were made:

i) Background concentrations were kept at the same values as 1994. Although (Artaxo 1998) have estimated long range contributions from copper smelters that will undergo emission reduction plans, those contributions are fairly modest for all PM fractions, so we are not considering smaller background for any of the future scenarios (either BAU or CP).

ii) The β coefficients include secondary aerosol production that ought to be related to the amount of NOx being released in the region, so this is a drawback of our current model. In the absence of further information, we will keep these parameters with the same values as in 1994. The estimations of CENMA (CENMA 1997) for the 2005 EIS indicate almost no change of the ratio of PM_{10} to SO_2 emissions coming from stationary sources.

iii) The α parameter was modified in the CP scenario to reflect improvements in the emission performance of mobile sources. Using estimates of emission factors for mobile sources in Chile (CENMA 1997), it was estimated that the ratio of future a values to the 1994 values would reach the following values: 0.95 by 2000, 0.86 by 2005, 0.75 by 2010, 0.70 by 2015 and 0.65 by 2020. This reduction was introduced in the model estimates under the CP scenario; for the BAU scenario no change in α was considered.

Figures 1 and 2 show the projected impacts of PM_{10} and $PM_{2.5}$, respectively, at the four MACAM stations A,B,C and D. The estimated impacts are the fall and winter annual average concentrations at those four monitoring sites; the continuous lines are drawn for the BAU scenario and the dashed lines correspond to the CP scenario. In both figures it is clear that by 2020 the two scenarios achieve different impacts, with CP concentrations being lower by 10 to 20 ($\mu g/m^3$).

Figure 3-2 Projections for PM10 Impacts at the MACAM Monitoring sites

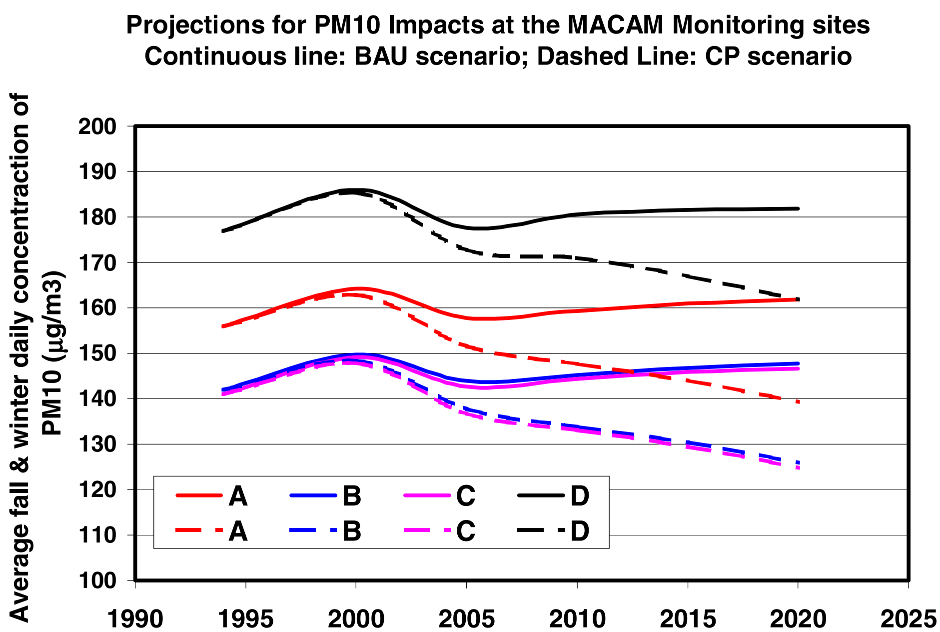
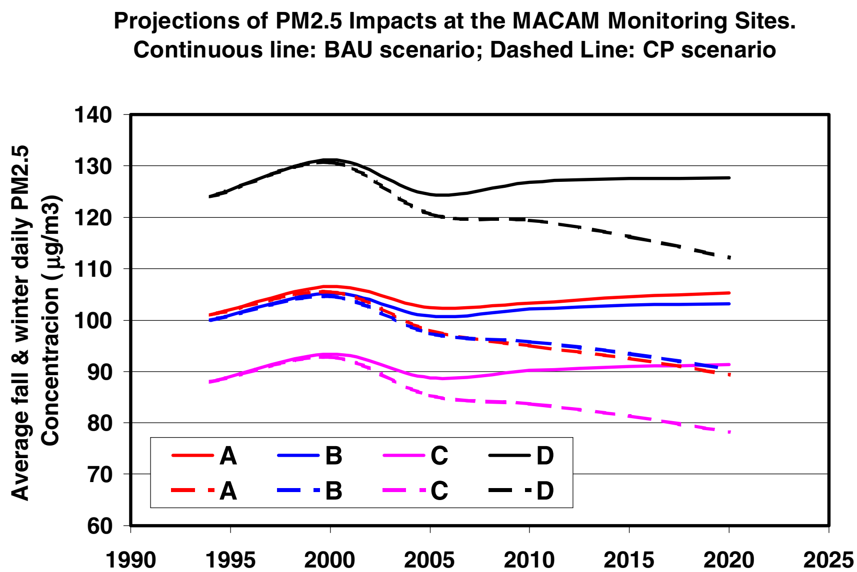


Figure 3-3 Projections for PM2.5 Impacts at the MACAM Monitoring sites



Certainly, this approach can be complemented in the future with the aggregated Latimer coefficients estimated for the four climatological regions in the USA by Pechan & Assoc. (E.H. Pechan & Associates 1997). In order to do this, the PRIEN data need to be revisited to express the scenarios in terms of energy consumption by type of fuel and type of economic sector; at this moment we are not sure if this is feasible. A second, long term option is to use the results of the CONAMA Central Chile air pollution modeling to assess the importance of secondary PM. Either of these two methods can be used to extrapolate changes in ambient air concentrations for the rest of the country, where few (if any) reliable data are available.

3.1.2 Method 2 : Source apportionment of fine particular matter concentrations

We estimated the changes in ambient PM concentrations due to changes in primary pollutant emissions using an alternative method. The method is based on source apportionment of $PM_{2.5}$ concentrations to primary pollutants conducted in Santiago in 1996 and 1998 (Artaxo 1996; Artaxo 1998; Artaxo, Oyola, and Martinez 1999). We estimated the fraction of $PM_{2.5}$ concentrations in Santiago due to each primary pollutant, based on those measurements, and obtained the fractions shown in the next table.

Figure 3-4 Percentage of $PM_{2.5}$ concentrations attributable to each primary pollutant in Santiago, 1998.

Primary Pollutant	Percentage attributable	95% confidence interval
Resuspended Dust	5%	(0.5% - 10%)
SO2	20%	(15.5% - 25%)
NMHC	0%	(0% - 0%)
NOx	30%	(21.1% - 39%)
PM10	34%	(24.6% - 42%)
Other	11.5%	- -
Total	**100.0%**	

Source: own estimates based on (Artaxo 1996) and (Artaxo, Oyola, and Martinez 1999).

In the above table PM10 should be understood as primary emission of PM, whereas SO2 and NOx are associated to secondary sulphates and nitrates, respectively.

If we assume that the contribution of each primary pollutant is constant over time, and that there are no interactions between pollutants, then the relative change in ambient $PM_{2.5}$ concentrations can be expressed as

$$\Delta\%[PM_{2.5}] = \sum_i F_i \cdot \Delta\%[P_i]$$

where

- $\Delta\%[PM_{2.5}]$ is the relative change in $PM_{2.5}$ concentrations
- $\Delta\%[P_i]$ is the relative change in pollutant i concentrations
- F_i is the fraction of $PM_{2.5}$ apportioned to pollutant i.

This equation should be applied only to the fraction of the PM concentration above the background concentrations. However, we should consider only the natural background, not the background due to emissions occurring elsewhere in the country (by this we mean secondary PM2.5 formation). In effect, if we are conducting an analysis for the whole country, assuming a relatively uniform distribution of pollutant sources within the country, the background concentration will also change when the level of emissions changes within the whole country.

The source apportionments of PM2.5 concentrations vary by region of the country. However, at this moment we do not have the necessary data to estimate it for each region. Therefore, we extrapolated the Santiago results to the whole country.

3.2 Health Impact Estimates

There is a growing number of studies linking particulate air pollution with both mortality and morbidity all over the world. For short tem effects, the work of Dockery and Schwartz in the late eighties has been replicated in more than 40 cities to date (and the number keeps growing), although still most of the studies come from US and European cities. For chronic effects, two prospective studies conducted in the US, the Harvard Six cities study (Dockery et al. 1993) and the Pope and colleagues study (Pope III et al. 1995) have shown significant results, in agreement with earlier results from Lave and Seskin (Lave and Seskin 1977). Although the causal mechanism by which particulate matter can induce death is not yet know, there is not much doubt than the association is not a spurious one, and several countries, including the US and the EU had moved towards more stringent standards based on the recent studies (EPA 1997b; WHO 1995)

For morbidity effects, studies in several countries have associated particulate matter with hospital admissions for several causes, emergency room visits, increased incidence asthma attacks, work loss days, restricted activity days, and minor symptoms, as well as increased incidence of chronic bronchitis (EPA 1996).

Most of the studies linking air pollution and health have used a Poisson model, in which the mean of the daily effects (Y) is modeled as an exponential function of the explanatory variables (X):

$$E(Y) = \exp[\beta * X]$$

In this model, the relative risk (RR) associated with a change in the PM concentrations (one of the X variables) is given by

$$RR(\Delta PM) = \exp[\beta * \Delta PM]$$

The slope coefficient, β, is obtained from the epidemiological studies, as shown later. ΔPM is the change in PM concentrations from a reference concentration. The relative risk needs to be applied to a base rate of effects, which is obtained from the observation of effects on the population that is exposed to some level of air pollution. Therefore, it is convenient to refer the change in pollution to the existing concentrations, C_O. For an increase in concentrations ΔPM from C_O to C, the change in effects is given by:

$$\mathrm{E}[\mathrm{Effects}(\Delta\mathrm{PM})] = [\exp(\beta * (C - C_0)) - 1] \cdot \mathrm{Effect\ rate}_0 \cdot \mathrm{Population\ exposed}$$

where $\mathrm{EffectRate}_0$ refers to the number of effects at concentration C_O, and is generally obtained from health statistics data. The above formula assumes that there is no threshold in the effects. If there is a threshold in the effects, i.e. a concentration C_T below which there are no effects, the formula becomes

$$\mathrm{E}[\mathrm{Effects}(\Delta\mathrm{PM})] = [\exp(\beta \cdot (C - \max\{C_O, C_T\})) - 1] \cdot \mathrm{EffectsRate}_0 \cdot \mathrm{Population\ exposed}$$

Since β is usually small, the above formula can be linearized using Taylor's expansion for the exponential function, obtaining simply βC instead of exp(βC).

For short terms studies, like the daily time series studies, the above formula applies to daily effects, and the effects rate should be expressed as the number of effects per day. To obtain the number of excess effects in a year, it is necessary to add the effects for all days of the year. If the effects rate and population exposed are constant throughout the year, we obtain:

$$\left[Effects/Year\right] = Pop_{\exp} \cdot EffectsRate \cdot \sum_{i=1}^{365-d} \beta \cdot \left|C_i - C_T\right|^{+} \quad with \quad \begin{cases} \left|C_i - C_T\right|^{+} > 0 \\ d = N^{o}\ of\ \ days\ where\ \ C_i \leq C_T \end{cases}$$

If all days in the year are above the threshold concentration C_T, then we can take β out the summation on the right, and the formula can be expressed in terms of the annual daily average concentration $\overline{C}$:

$$\left[Effects/Year\right] = Pop_{\exp} \cdot \mathrm{EffectRate}_0 \cdot \beta \cdot \overline{C}$$

It is important to mention that when there is a threshold in the effects, this is always an approximate formula, even if the annual daily mean is above the threshold level. In general, a fraction of the days of the year will be below the threshold concentration. For computing the exact number of effects in this case it is necessary to know the form of the distribution of the daily concentrations. Generally, it is assumed that daily concentrations are distributed lognormal (Ott 1990), although other distributions have been shown to better represent the physical process underlying air pollution concentrations [Morel, 1999 #1199

Base rate of effects. The other parameters needed to compute the total number of effects are the exposed population and the effects base rate. We projected the exposed population using the estimates of the Chilean Institute of Statistics, considering that the age distribution remains constant. For the base rate of the effects we used the rates for Santiago for all the cities, and assumed that they were constant in time.

Exposure-response functions. We conducted the analysis based on exposure-response functions obtained from the literature, mainly from the estimation of benefits of the Clean Air Act performed by EPA (EPA 1997a) and from the recommendations of the World Health Organization by Ostro (Ostro 1996). We complemented these sources with exposure response functions from studies performed in Santiago. For mortality we used (Ostro et al. 1996) and our own studies (Cifuentes, Vega, and Lave 1999). For child medical visits, we used (Ostro et al. 1999) and (Illabaca et al. 1999). All of the studies correspond to short-term effects, except for chronic bronchitis and mortality. The estimate for chronic mortality effects from the studies of Dockery at al (Dockery et al. 1993; Pope III et al. 1995) are quite high. Following Ostro 1996, we included the point estimate of these studies (3 % increase per 10 μg/m^3 of PM_{10}) only for the high case of mortality. Whenever possible, we used exposure-response based on $PM_{2.5}$ If they were available for PM_{10}, we convert them to $PM_{2.5}$ using the relation $PM_{2.5} = 0.55\ PM_{10}$.

We considered three age groups in the analysis: Children 0-18 yrs, Adults, 18-64 yrs, and 65+ yrs, In some cases, we considered specific age groups, like for asthma attacks, in which the exposure-response functions are for children below 15 yrs., or consider the whole population, as for mortality effects. The summary of the exposure-response coefficients for the effects considered is shown in the next table.

Figure 3-5 Summary of exposure response coefficients used in the analysis

Endpoints	Age Group	Pollutant	Mean	t stat	Source
Chronic Mortality	All	PM2.5	0.00450		Ostro 1996 (high case)
Chronic Bronchitis	> 65 yrs	PM10	0.02100	4.2	Schwartz et al,1993
Premature Deaths (short term)	All	PM2.5	0.00120	3.9	Own analysis
Hospital Admissions RSP	> 65 yrs	PM10	0.00169	3.8	Pooled
Hospital Admissions COPD	> 65 yrs	PM10	0.00257	6.4	Pooled
Hosp. Adm Congestive heart failure	> 65 yrs	PM10	0.00098	3.2	Schwartz & Morris, 1995
Hosp Adm Ischemic heart failure	> 65 yrs	PM10	0.00056	2.7	Schwartz & Morris, 1995
Hospital Admissions Pneumonia	> 65 yrs	PM10	0.00134	5.1	Pooled
Asthma Attacks	All	PM10	0.00144	4.6	Ostro et al, 1996
Acute Bronchitis	Childs	PM2.5	0.00440	2.0	Dockery et al., 1989
Child Medical Visits LRS	Childs	PM10	0.00083	2.5	Ostro et al, 1999
Emergency Room Visits	All	PM10	0.00222	5.2	Sunyer et al, 1993
Shortness of Breath (days)	Childs	PM10	0.00841	2.3	Ostro et al, 1995
Work loss days (WLDs)	18-64 yrs	PM2.5	0.00464	13.2	Ostro et al, 1987
RADs	18-64 yrs	PM2.5	0.00475	16.5	Ostro et al, 1987
MRADs	18-64 yrs	PM2.5	0.00741	10.5	Ostro et al, 1989

3.3 Effect Valuation

To estimate the social benefits due to reduced health effects, it is necessary to estimate society's losses due to the extra occurrence of one effect. Several methods exist to value such losses. The most straightforward one is based on the direct losses to society, stemming from the cost of treatment of each effect plus the productivity lost. This approach, which is known as the human capital method for valuing mortality effects, and the cost of illness approach for valuing morbidity effects, suffers from a serious limitation, by not considering the disutility suffered by the individual affected and by society as a whole. However, due to its relative ease of calculation, it has been used in previous analysis of quantification of air pollution effects, such as the economic valuation of the benefits associated to the Decontamination Plan of Santiago (Comisión Nacional del Medio Ambiente 1997).

We used two sets of values for our analysis. As a starting point, we took the values derived previously by Conama for the analysis of the social benefits of the Decontamination Plan of 1997, updated for the year

2000 by the growth of per capita income. As mentioned before, these are mostly based on the human capital or cost of illness methods . This scenario was called “PPDA values”, for the acronym of the Decontamination Plan.

Our own set of values for each effect is based on values used by the USEPA (EPA 1997a), that are mostly based on WTP. We transferred the values from the U.S. to Chile using the ratio of the per capita income of both countries, which was five to one in 1995. By far, the more important effects are premature mortality. For these effects, we choose a lower bound from the range of values used by EPA, which became US$375 thousand after adjustment for year 1997. This value falls within the range of values that we have obtained in a pilot test of a contingent valuation study of willingness to pay for reducing mortality risks in Santiago (Cifuentes, Prieto, and Escobari 1999). When the results of the full survey are available, earlier next year, they will be incorporated into the analysis. The summary of values used in the analysis in shown in the next table.

The values were updated annually using a projected growth in real per capita income of 4.5%. For example, in 2020 a premature death averted has a value of US$ 983 thousands (in 1997 dollars)

Figure 3-6 Unit values for each effect for each valuation scenario for year 2000 (1997US$ per effect)

		Valuation Scenario	
Endpoint	Age Group	Own	PPDA
Chronic Mortality	All	407,798	69,497
Chronic Bronchitis	> 65 yrs	68,001	83,449
Mortality	All	407,798	69,497
Hospital Admissions RSP	> 65 yrs	3,191	721
Hospital Admissions COPD	> 65 yrs	4,106	721
Hosp. Adm Congestive heart failure	> 65 yrs	4,342	721
Hosp Adm Ischemic heart failure	> 65 yrs	5,387	721
Hospital Admissions Pneumonia	> 65 yrs	4,158	721
Asthma Attacks	All	8	11
Acute Bronchitis	Childs	11	16
Emergency Room Visits	Childs	60	47
Child Medical Visits	All	183	0
Shortness of Breath (days)	Childs	1	2
Work loss days (WLDs)	18 - 64 yrs	22	21
RADs	18 - 64 yrs	10	8
MRADs	18 - 64 yrs	9	8

3.4 Uncertainty and variability analysis

As has been shown in the preceding sections, each step of the analysis is fraught with uncertainty. Explicit consideration of all the uncertainties is crucial to illuminate the analysis for several reasons (Morgan and Henrion 1990):

- It let us identify the important factors in the analysis
- It can help us identify which steps of the analysis need to be improved the most
- It points out potential sources of disagreement between different experts or analysts

Uncertainty comes from several sources. One possible classification is into parameter uncertainty, model uncertainty, and scenario uncertainty. In this analysis, we have considered explicitly only the first two. Parameter uncertainty can be modeled quantitatively treating the parameters as stochastic variables. We have done so for the exposure-response coefficients for health effects quantification, and for some parameters of the ambient concentration models. Other variables have been treated parametrically, such as society's willingness to pay to avoid an extra health effect. For this, we have considered two different sets of values. A more difficult kind of uncertainty is model uncertainty. We have considered two different models to estimate the change in $PM_{2.5}$ concentrations due to changes in emissions. Finally, scenario uncertainty has not been considered in this analysis yet. A sure candidate for this is the mitigation scenario. External factors, like the policies taken by Chile, can change this scenario dramatically.

Variability refers to another kind of uncertainty, in which the results of the analysis depend on the geographical and temporal resolution of the model used (Frey and Rhodes 1996). Consideration of variability is crucial in this analysis, as has been shown by Wang and Smith (Wang and Smith 1998). At this moment however, our analysis is performed at an aggregate level, so we are missing potential variability in the results.

To consider quantitatively the uncertainty in the analysis, a model was implemented in the Analytica modeling environment (Lumina Decision Systems 1998). This very flexible modeling environment let us propagate and analyze the uncertainty of the parameters and the results.

4. Results

Based on the emissions changes presented in the first section, we estimated the evolution of $PM_{2.5}$ concentrations in time for both methods proposed. The next table shows the point estimates for each method.

Figure 4-1 PM2.5 concentrations relative to year 2000 concentrations, for both methods of estimating the concentrations

Year	Method for estimating PM2.5 concentrations					
	Box Model			Source apportionment		
	BAU	CP	BAU - CP	BAU	CP	BAU - CP
2000	1	1	0	1	1	0
2005	0.95	0.93	0.02	1.02	1.01	0.01
2010	0.95	0.91	0.03	1.08	1.06	0.02
2015	0.98	0.89	0.09	1.14	1.09	0.06
2020	0.98	0.86	0.12	1.20	1.12	0.08

The table shows that even both methods produce quite different results for each scenario, BAU and CP. For the box model, concentrations are decreasing in the future, due mainly to the decrease in SO_2 emissions. For the source apportionment method, concentrations increase, driven mainly by the increase in NO_x emissions. However, the difference between the BAU and CP scenarios is not that different for each method. The box-model method shows a bigger concentration reduction for the CP scenario than the source apportionment method.

Applying the changes in $PM_{2.5}$ concentrations to the exposed population in each city it is possible to compute the excess health effects. The next table shows the excess health effects in the year 2020 for each of the policy scenarios. The excess effects have been computed assuming there is no threshold in any of the effects. The table shows the mid value of the effects for each policy scenario, grouped by type of effect, for all age groups considered. We have grouped the effects by type of effect into more aggregated categories.

Figure 4-2 Mid value of total excess effects in the year 2020 for each policy scenario,

Endpoint	Policy Scenario BAU	CP	BAU - CP
Premature Death	2,681	2,349	332
Chronic Bronchitis	13,865	12,147	1,719
Hospital Admissions	16,083	14,090	1,993
Emergency Room Visits	212,954	186,559	26,395
Child medical visits	97,165	85,122	12,043
Asthma Attacks & Bronchitis	2,549,017	2,233,077	315,940
Restricted Activity Days	53,870,280	47,193,277	6,677,003

Note: mid value estimates for $PM_{2.5}$ concentration changes estimated using the box-model method

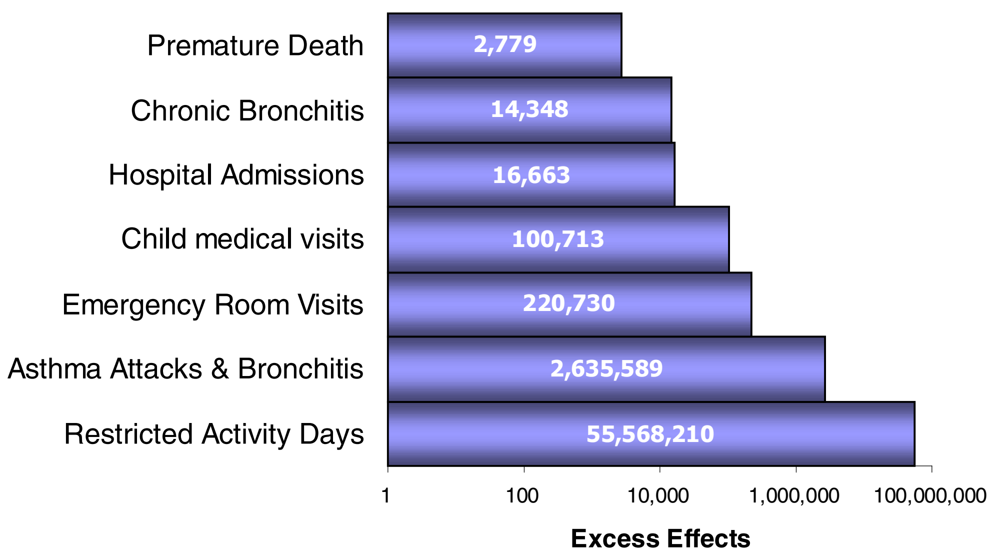
Given the schedule of emissions mitigation, this is the maximum number of effects avoided for the period of analysis. The next table shows the total number of effects avoided from 2000 to 2020 for the CP scenario compared to the BAU. The table shows the mid estimate, and the 95% confidence interval, computed using Montecarlo simulation (using the Analytica modeling environment).

Figure 4-3 Total effects avoided in the CP scenario with respect to the BAU scenario during the period 2000 to 2020.

Endpoint	Number of effects avoided mid	95% confidence interval
Premature Death	2,779	(1,703 - 9,074)
Chronic Bronchitis	14,348	(11,953 - 15,730)
Hospital Admissions	16,663	(13,109 - 21,573)
Emergency Room Visits	220,730	(160,734 - 274,918)
Child medical visits	100,713	(38,195 - 157,983)
Asthma Attacks & Bronchitis	2,635,589	(1,849,734 - 3,390,035)
Restricted Activity Days	55,568,210	(39,103,270 - 65,650,670)

Note: $PM_{2.5}$ concentration changes estimated using the box-model method

Figure 4-4 Total effects avoided in the CP scenario with respect to the BAU scenario during the period 2000 to 2020.



For the whole period of analysis, the mid estimate is around 2,800 deaths can be avoided, with a 95% confidence interval of 1,700 to 9,100. The upper bound of this interval is much higher because it includes chronic effect deaths. Most of these effects will occur in the Metropolitan Region of Santiago. Using the unit values shown in the preceding chapter, we computed society's social losses due to these health effects. The difference of the damages for each scenario is the social benefit of the mitigation measures. We have computed the present value of this benefits for the whole period, using a real discount rate of 12%, which is the rate used in Chile for evaluation of all social projects

Figure 4-5 Mid value of social losses for each scenario for the year 2020 (Millions of US$)

Endpoint	Policy Scenario BAU	CP	BAU - CP
Premature Death	1,795.7	1,468.8	326.8
Chronic Bronchitis	1,548.5	1,266.7	281.8
Hospital Admissions	109.1	89.2	19.9
Emergency Room Visits	21.2	17.3	3.9
Child medical visits	29.1	23.8	5.3
Asthma Attacks & Bronchitis	35.3	28.8	6.4
Restricted Activity Days	512.0	418.8	93.2
Total	**4,051**	**3,314**	**737**

Note: $PM_{2.5}$ concentration changes estimated using the box-model method. Social losses computed using our own values.

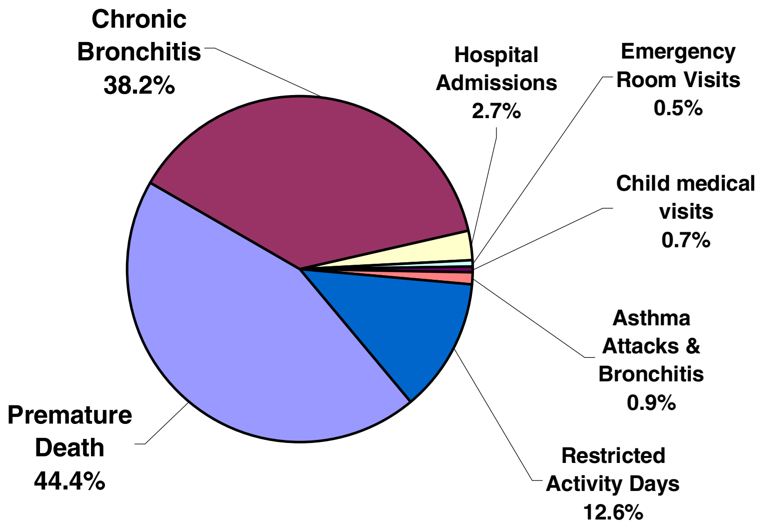
Figure 4-6 Present value of social benefits of the CP scenario with respect to the BAU for the period 2000 to 2020 (Millions of US$)

Endpoint	mid	95% CI
Premature Death	443.6	(271.9 - 1,448.6)
Chronic Bronchitis	381.7	(316.5 - 428.8)
Hospital Admissions	26.9	(22.1 - 32.8)
Emergency Room Visits	5.2	(3.8 - 6.5)
Child medical visits	7.2	(2.7 - 11.3)
Asthma Attacks & Bronchitis	8.6	(6.1 - 11.1)
Restricted Activity Days	126.0	(115.9 - 136.3)
Total	**999.2**	**(739.1 - 2,075.5)**

Note: $PM_{2.5}$ concentration changes estimated using the box-model method. Social losses computed using our own values. Present value computed using a 12% real discount rate.

Where do these benefits come from? The next figure shows the share of the present value of the benefits for each effect. It is clear that the biggest share of the benefits comes from avoided premature mortality, although chronic bronchitis cases also have an important contribution in the mid value case. Premature mortality dominates the values for the upper bounds of the confidence interval, representing around 70% of the benefits. This is mainly due to the consideration of chronic effect deaths for this scenario.

Figure 4-7 Share of the present value of benefits for each type of effect (mid estimates)



Note: For ouwn values scenario

All the previous results have been obtained using the box-model to estimate the change in $PM_{2.5}$ concentrations, and our own valuation scenario. The next table shows the net present value of the benefits for the other set of values and the other model for computing the changes in $PM_{2.5}$ concentrations.

Figure 4-8 Present value of social benefits of CP vs BAU for years 2000 to 2020 for each valuation scenario and each method of emissions impacts estimation (Million of US$)

Valuation Scenario	Method for estimating PM2.5 concentrations			
	Box Model		Source apportionment	
Own Values	999.2	(739.1 - 2,075.5)	604.8	(463.6 - 1,311.7)
PPDA Values	682.4	(599.0 - 881.8)	415.1	(338.8 - 599.8)

Notes:
the upper bound of the CI for mortality corresponds to the upper value of the acute effects plus the mid value of the chronic mortality effects)

Another way to look at these results is to compute the social benefit accrued from the reduction of each ton of CO_2 equivalent. This is done by just dividing the benefits due to avoided health effects in each year by the CO_2 reductions obtained that year. The next table shows the results for all the years of analysis.

Figure 4-9 Social Benefit per ton of CO2eq reductions for each year (US$ / ton CO2equ-year)

Years	Box Model		Source apportionment	
	Own Values	PPDA Values	Own Values	PPDA Values
2010	20.8 (17 - 42)	14.2 (13 - 19)	12.8 (9 - 29)	8.7 (6 - 13)
2020	50.4 (41 - 103)	34.6 (31 - 45)	33.4 (26 - 72)	22.6 (18 - 33)

5. Discussion

This work is a preliminary estimation of the potential co-control benefits of greenhouse mitigation measures in Chile. We have conducted an aggregate analysis for the whole country, based on previously developed base (BAU) and mitigation (CP) scenarios.

The results show potentially high co-control social benefits. The implementation of the CP scenario may prevent 2,800 deaths in the period 2000 to 2020, with a range from 1,700 up to 9,100. The mid estimate rests on generally accepted concentration-response coefficients, which are in agreement with studies conducted in Santiago, Chile's capital, which accounts for most of the exposure to particulate matter. The upper bound of the confidence interval relies heavily on the mortality estimates from prospective studies performed in the US, under different conditions than in Chile, so their application is more uncertain.

From an economic point of view, the potential co-control benefits represent a substantial fraction of the potential costs of the mitigating options. For 2010, the benefits per ton of CO_2 abated range from 6 up to 42 dollars, depending on the models used to estimate the impact of emissions on concentrations, and the scenarios considered to value the effects. For 2020, the values range from 18 to 103 dollars per ton abated. The magnitude of these values is comparable to the current estimates of abatement costs for CO_2, for normal mitigation scenarios. Therefore, these co-benefits can offset a big fraction of the costs needed to implement the measures. In the specific case studied here, where the mitigation scenario corresponds to a "no-regrets" case, in which all the measures considered do not impose a cost on the user, these co-benefits signal a net benefit for society.

However, it is necessary to stress that this is a preliminary analysis, which suffers from many limitations. The main one is that it has been conducted at an aggregate level for the whole country, with no consideration of local conditions, like emissions locale, meteorology or population density close to the source. Therefore, our estimates are 'average' estimates. Several factors can influence the analysis, making the impact of the emissions vary widely. For example, Wang and Smith (Wang and Smith 1999) have shown that the impact of a ton of PM_{10} emitted in a power plant or a ton of PM_{10} emitted in household stoves can vary by a factor of 40, due mainly to the greater indoor exposure associated with the latter. Consideration of these factors is crucial to estimate the co-benefits associated with specific measures.

Our future work focus on developing the models to analyze in detail a mitigating measure, considering all the factors that may influence its impact, and on refining our existing analysis framework.

Among the models we want to develop are better models to estimate the exposure to fine particulate matter resulting from changes in local emissions, taking into account local conditions and meteorology. We are also working on emissions scenarios with higher resolution both in the sectors considered in the previous work, and in the type of fuel used in each sector. This improved resolution will help us consider sector and fuel specific emissions factors, developing more precise estimations of primary pollutant emission changes, but more important, giving better temporal and geographical resolution of the emission changes. These can help us estimate the changes in exposure associated to these emission changes with more detail.

With respect to our general framework of analysis, we are planning several improvements, based on our own results from a time-series study in Santiago, and from our contingent valuation survey in Santiago. First, we plan to include more age groups, especially for older ages, considering the impact of age on the risk and on the willingness to pay to reduce it. We will also investigate the inclusion of effects for children less than one year of age, which have been found in Mexico City (Loomis et al. 1999), but which we have not yet confirmed in Santiago.

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